9 WATERSHED LAND USE

Stream conditions are often influenced by human activities in the surrounding watershed. Historically, much of Maryland was covered by forest, a sharp contrast to the variety of urban and agricultural uses presently dominating the landscape (Figure 3-5). Current stream conditions are in part determined by these human uses of watershed lands. Results in this chapter describe the range of land uses in watersheds upstream of sites sampled in the Maryland Biological Stream Survey (MBSS, or Survey) and explore the associations between land use and stream conditions, using biological and physical habitat indicators.

9.1 BACKGROUND

Human activities affect streams at a variety of spatial scales. Rivers are by nature hierarchical systems, so the character of a local stream site is to some degree controlled by the larger-scale river system and watershed to which it belongs. This means that to fully understand the multiple, cumulative impacts on stream systems, conditions at a broad landscape scale, as well as the local or site-specific scale, must be assessed. For example, while water chemistry results may indicate that acidic deposition is the likely cause of degraded fish communities at a particular site, there may be other stresses on that stream that would continue to inhibit fish or other stream biota even if the acidification was ameliorated. Urban development and the clearing of riparian vegetation upstream of the site may also be causing hydrological changes that accelerate bank erosion and sedimentation. In other cases, refugia within a local stream network may mitigate severe episodic stresses. This illustrates the need to include landscape-level information in the ecological assessment process. Only by using an integrated multiple-scale approach can the Survey provide context for evaluating the relative contributions of different anthropogenic activities.

One measure of anthropogenic influence at the landscape scale is watershed land use. Watersheds form natural geographic units for assessing impacts on streams, because land use within the watershed (or catchment) upstream of a specific stream site is representative of many of the human activities affecting the stream at that point. As such, land cover serves as a surrogate for a variety of stressors, some of which may be difficult to measure directly.

Because no field sampling program will ever be able to visit all sites or all streams through the state, the "wall-to-wall" coverage provided by land cover data serves as a useful tool for predicting conditions at sites that might otherwise be overlooked. Geographic information system (GIS) data may be used to develop predictive models linking land cover with instream biological or physical habitat conditions. In evaluating streams across a large area, GIS land cover information can be employed in an initial screening step to locate areas most likely to exhibit desirable or degraded conditions and to then target subsequent field sampling to these streams. Depending on management goals, these more detailed investigations would provide information needed to make decisions about appropriate conservation or restoration actions.

In much of the United States, conversions of naturally vegetated watershed lands to urban and agricultural uses have resulted in serious impacts to streams and their aquatic inhabitants. Examining land uses as stressors, through analyses of relationships with ecological indicators, allows predictions to be made about the extent and severity of ecological impacts associated with varying levels of human use. Some investigations have indicated that development of even small portions of a watershed may affect stream biota. For example, impervious surface covering 10-20% of the watershed area can have detrimental effects on streams (Schueler 1994). Impervious surfaces, such as roads, parking lots, sidewalks, and rooftops, cause a rapid increase in the rate at which water is transported from the watershed to its stream channels. Effects include more variable stream flows, increased erosion from runoff, habitat degradation caused by channel instability, increased nonpoint source pollutant loading, elevated temperatures, and losses of biological biodiversity.

Reviews of stream research in numerous watersheds (Center for Watershed Protection 1998, Schueler 1994) indicate that impacts on stream quality are commonly noted at about 10% coverage by impervious surface. Effects on sensitive species may occur at even lower levels (see brook trout example in Section 4). With even more impervious surface, most notably at about 25-30% of catchment area, studies have shown that numerous aspects of stream quality become degraded, including biological integrity, water quality, and physical habitat quality (Center for Watershed Protection 1998).

In this section, we examine urban land use, which represents impervious surface and other aspects of urbanization that affect stream quality. Note that the percent coverage by impervious surface for a catchment would be lower than the corresponding value for percent urban land assessed by the Survey. According to the class definitions used in developing the land cover base data (MRLC 1996 a,b), impervious surfaces make up 30-80% of the low-intensity and 80-100% of high-intensity developed urban land classes. Other land cover classes contribute smaller but possibly significant proportions of impervious surface. Therefore, the values for percent urban land use associated with poor stream quality were expected to be somewhat higher than the 10-30% impervious surface threshold reported by others.

Associations between urban or agricultural watershed land use and stream biota have been examined in a number of studies (e.g., Klein 1979, Steedman 1988, Richards et al. 1996, Roth et al. 1996). In this chapter, we report on the relationships observed between land use and several indicators of stream condition for sites sampled in the 1995-1997 MBSS. Ecological indicators included the fish Index of Biotic Integrity (IBI), benthic macroinvertebrate IBI and Hilsenhoff Biotic Index, number of aquatic salamander species, and Physical Habitat Index (PHI). Because the Survey employs a probability-based design, examining land use associations for the sampled sites allows us to make inferences about the effects of land use on biological resources statewide and within individual basins.

9.2 CHARACTERIZATION OF LAND USE IN UPSTREAM CATCHMENTS

A characterization of catchment land use was developed for the watershed upstream of each site sampled in the 1995-1997 MBSS using the GIS methods described in Chapter 2. Statewide, the dominant land use in site-specific catchments was forest (mean percent cover of 46%), followed by agriculture (44%) and urban (9%). In individual basins (Figure 9-1), agricultural land use was greatest at sites in the Susquehanna basin, with a per-site mean of 66%. Agriculture also dominated in the Middle Potomac, Gunpowder, and Elk basins, all with a per-site average of 63%. Sites in the North Branch Potomac had a mean of just 15%, while the mean in the remaining basins ranged from 22 to 60% agricultural land. Forest cover was most extensive for sites in the North Branch Potomac basin (83%) and least extensive in the Patapsco basin (1996 sampling, 21%). As expected, urban land use was greatest in the Patapsco (1996 sampling, 31%) and Potomac Washington Metro (23%) basins. Four of the remaining basins: the Patuxent, West Chesapeake, Patapsco (1995 sampling), and Bush basins contained a mean percentage of urban land use between 15 and 20%. The remaining basins had a mean percentage of urban land use that was less than 10%.

9.3 EXAMPLES OF LAND USE EFFECTS ON STREAM WATER QUALITY

One way that urbanization can affect stream water quality is through changes in water temperature. Stream water temperature is greater and more variable in streams draining urban lands than in streams draining forest lands. During summer, rain running off of hot impervious surfaces (parking lots, rooftops, etc.) and directly into streams causes temperature spikes during storms. Also, urban watersheds are likely to be less shaded than more natural forested watersheds. Where impervious surface is extensive, reduced infiltration may result in reduced groundwater input to stream baseflow. All of these factors contribute to higher average water temperatures and larger spikes in urban watersheds relative to forested watersheds.

In the Patuxent basin, during 1997, water temperature was measured at all MBSS sites every 15 minutes by continuous temperature loggers from June 5 to September 15. Mean daily temperatures ranged from 17°C (63°F) to 23°C (73°F), with an overall mean of 20°C (68°F). The maximum temperature reached in any stream was 31°C (88°F). Thus no sites in the basin exceeded the State Use I Temperature Criterion of 32°C (90°F) (COMAR 1995).

Two streams in the Patuxent basin illustrate the differences in stream water temperature based on the percentage of urban land in the catchment. Dorsey Run and Midway Run are second-order Coastal Plain streams with similar widths and depths (at the sampling sites) but fairly different land uses (Figure 9-2). Dorsey Run's watershed is mostly forested (73%), with only 10% urban land. The remainder of its watershed (17%) is agricultural. Midway Run's watershed, however, is nearly evenly split between forest (32%), urban (37%), and agricultural (31%) land.

During July 1997, the water in Midway Run was warmer in the daytime (and cooler at night) than Dorsey Run (Figure 9-3). Also, the highest daytime temperatures were reached more quickly in Midway Run than in Dorsey Run. The comparison between these two watersheds demonstrates how the loss of natural land cover can negatively affect water quality and potentially impair aquatic life, even though no regulatory criteria are exceeded.

Another way land use affects stream water quality is illustrated by the relationship between agricultural land use and instream nitrate-nitrogen (NO_3-N) concentration.

MBSS sites were divided into two groups: those with catchments dominated by agricultural land uses (>50% agriculture) and those with catchments predominately in other land uses (<50% agriculture). Spring baseflow NO_3-N concentrations were compared between the two groups. Among sites with >50% agriculture, the statewide mean NO_3-N concentration was 4.0 mg/l, more than three times the mean NO_3-N concentration among sites with <50% agriculture (mean NO_3-N of 1.2 mg/l). Within nearly every individual basin, NO_3-N concentrations were substantially higher among sites with agriculture >50% (Figure 9-4).

9.4 ASSOCIATIONS BETWEEN LAND USE AND ECOLOGICAL INDICATORS

9.4.1 Associations Between Land Use and the Fish IBI

For sites sampled in the 1995-1997 MBSS, fish IBI scores were plotted against the percentage of catchment area in various land uses (e.g., urban, agricultural, forest). Linear regression analyses were conducted to evaluate the strength of associations between land use and biological condition.

For all basins combined, fish IBI scores decreased with increasing urban land use (Figure 9-5; $p < 0.001$, r^2=0.09). Nearly all sites with greater than 50% of the catchment in urban land use had IBI scores indicating poor to very poor conditions (i.e., IBI < 3.0). However, among sites with a lower percentage of urban land use, a wide range of IBI scores was observed, representing good to very poor conditions. This suggests that factors other than urbanization have a strong influence on biological condition at these sites. Fish IBI showed a significant negative correlation to increasing urban land use in two of the individual basins: the Potomac Washington Metro (Figure 9-6; r^2=0.24) and the Patapsco (Figure 9-7; r^2=0.63). Catchments in these two basins have the largest amount of urban land area (average land use of 31% and 23%, respectively). Statewide, they also account for many of the sites that contain more than 50% urban land. Many of the remaining basins have very few sites with more than 25% urban land. In fact, there are several basins that have no sites with more than 10% urban land. These sites probably fall below the level at which significant effects of urbanization could be detected at this scale of analysis. In these less urbanized basins, factors other than urbanization appear to more strongly influence the degradation of stream quality.

The associations between fish IBI and more specific urban land use categories paralleled the general fish IBI and urban land use relationship. For many sites, the majority of urban land was characterized by low-intensity development, including areas with a mixture of built structures and vegetation. This is common in suburban neighborhoods dominated by single-family housing. The intensity of low-intensity developed areas ranged from 0 to 87% of the watershed area for sampled sites. Overall, a smaller percentage of watershed areas were characterized by high-intensity development, including heavily built-up urban centers and large developments in suburban and rural areas. This category contains areas in which a significant land area is covered by concrete, asphalt, or other artificial materials, including apartment complexes, skyscrapers, shopping centers, factories, industrial complexes, airport runways, and interstate highways. The percentage of high-intensity developed areas ranged from 0 to 28% of the watershed area for sampled sites.

As with urban land use in general, fish IBI scores showed a significant decrease with low- intensity developed areas, both over all basins (Figure 9-8; $p < 0.001$, r^2=0.09) and within the Potomac Washington Metro (r^2=0.25) and Patapsco (r^2=0.63) basins. These two basins have the greatest number of sites with a high percentage of land (>25%) in low-intensity development. These results suggest that even less dense urbanization may have a significant effect on streams in certain areas. Fish IBI was also significantly correlated with high-intensity development over all basins ($p < 0.001$, r^2=0.08), even though there were few sites with greater than 25% of the catchment in high-intensity development.

For all basins combined, fish IBI scores showed a significant positive relationship with percentage of agricultural land, although there was a high degree of variability (Figure 9-9; $p < 0.001$, r^2=0.07). This relationship was also seen in six of the individual basins: the Potomac Washington Metro, West Chesapeake, Patapsco, Gunpowder, Chester, and Nanticoke/Wicomico (r^2=0.08-0.57). The Gunpowder basin effectively demonstrates this relationship between the percentage of agricultural land and the fish IBI (Figure 9-10; $p<0.002$, r^2=0.25). Several factors might explain why fish IBI scores increase with the percentage of agricultural land use. Foremost may be the fact that as the amount of agricultural land use in a given area increases, the amount of urban land cover (a factor likely to cause more pronounced stream degradation) will usually decrease. There are also many complex interactions between agricultural activities and responses in stream biota that may affect the fish IBI in different ways. For example, while agriculture may cause erosion and degrade fish habitat, runoff may contribute lime (which can neutralize acidic inputs) and nutrients (which can enhance stream productivity). In general, because agriculture is so pervasive throughout the state, it may be difficult to detect its effects within the range of impact assessed by the IBI.

To investigate differences in the effects of row crop agriculture and less intensive agricultural land use (such as hayfields and pastureland), the agricultural land use class was further subdivided into these two categories. As with agricultural land use in general, fish IBI scores increased with an increasing percentage of land use in both categories over all basins combined, with row crop agriculture showing a slightly stronger relationship with the fish IBI ($p < 0.001$, $r^2=0.10$). However, it was difficult to discriminate the effects of row crop agriculture from hay/pasture land because the two cover types tended to be correlated.

Forest land use, although often extensive, had no significant relationship to fish IBI scores statewide. One confounding factor was the impact of acid deposition and acid mine drainage on streams in forested watersheds. A number of sites with > 50% forest cover were affected by acid deposition and mine drainage, and many of these sites had fish IBIs lower than would be expected (Figure 9-11).

The percentage of catchment area as wetlands showed no significant relationship to fish IBI statewide. Wetlands effects may be particularly hard to detect, given that wetlands cover only a small percentage of land throughout the state. Among all sites sampled, wetlands made up only 0-5% of catchment land cover.

Sites with high fish IBI scores represent biological communities least affected by degradation and provide an additional basis for analyzing land use associations with stream condition. Sites with high fish IBI scores (i.e., those rated as good, IBI $\geq$ 4.0) were distributed throughout the state, as seen in the maps in Chapter 5. Generally, these streams were characterized by less urban development. Sites with IBI $\geq$ 4 had an average of 4% urban land use, compared with an average of 9% for all sites. This result emphasizes the large effect that urban development may have on stream water quality.

9.4.2 Associations Between Land Use and the Benthic Macroinvertebrate IBI

For sites sampled in the 1995-1997 MBSS, benthic IBI scores were plotted against the percentage of catchment area in various land uses (e.g., urban, agricultural, forest). Linear regression analyses were conducted to evaluate the strength of associations between land use and biological condition.

Statewide, benthic IBI scores decreased with increasing urban land use (Figure 9-12; $p < 0.001$, $r^2=0.17$). Nearly all sites with greater than 30% of the catchment in urban land use had benthic IBI scores indicating poor to very poor conditions (i.e., IBI < 3.0). This may suggest that the benthic IBI is more sensitive to an increase in urban land use than the fish IBI, which, on average, reached the threshold for poor condition at about 50% of urban land use. Benthic IBI scores were also negatively correlated with urban land use in six individual basins: the North Branch Potomac, Potomac Washington Metro, Lower Potomac, Patuxent, Patapsco, and Bush ($r^2=0.10$-0.44). The relationship of the benthic IBI to urban land use is shown for the Potomac Washington Metro basin (Figure 9-13; $r^2=0.44$) and for the Patuxent basin (Figure 9-14; $r^2=0.32$).

The relationship of benthic IBI to low-intensity development parallels that of urban land use in general, showing a significant decrease over all basins combined (Figure 9-15; $p < 0.001$, $r^2=0.16$). As with the fish IBI, these results show that even a small amount of development may drastically affect the quality of a stream. Benthic IBI was also significantly negatively correlated to high-intensity development over all basins sampled ($p < 0.001$, $r^2=0.15$), although very few sites contained a large amount of high-intensity development.

Statewide, benthic IBI scores were not significantly correlated with the percentage of land that is agricultural (Figure 9-16; $p<0.24$). This may indicate that the benthic IBI is a better indicator of degradation from urban land use than from agricultural land use. There are several reasons that the relationship of the benthic IBI to agricultural land use is not significant, including the confounding interactions with biota mentioned when discussing the fish IBI.

The relationship between benthic IBI scores and the percentage of the catchment as forested land was positive and significant statewide (Figure 9-17; $p < 0.001$, $r^2=0.06$). Sites affected by acid deposition and acid mine drainage, most having > 50% of the catchment as forest, resulted in some lower-than-expected benthic IBI scores. When these sites were excluded from analysis, the relationship was slightly stronger ($r^2=0.08$). Basins showing significant relationships between forest cover and benthic IBI scores were the Upper Potomac ($r^2=0.07$) and Patapsco ($r^2=0.06$). Because wetland areas made up such a small percentage of catchment land, there was no significant relationship between wetland land use and the benthic IBI ($p<0.74$).

9.4.3 Associations Between Land Use and the Hilsenhoff Biotic Index

The Hilsenhoff Biotic Index is a macroinvertebrate indicator of organic pollution tolerance (Hilsenhoff 1977, 1987, 1988). High scores are associated with pollution tolerant organisms and therefore with stream degradation. For sites sampled in the 1995-1997 MBSS, Hilsenhoff Biotic Index scores were plotted against the percentage of catchment area in various land uses, especially urban and agricultural. Linear regression analyses were conducted to evaluate the strength of associations between land use and biological condition.

Statewide, Hilsenhoff Biotic Index scores increased with increasing urban land use, indicating increased degradation with an increase in urban land (Figure 9-18; $p < 0.001$, $r^2=0.11$). This relationship was also significant in three of the basins: the Potomac Washington Metro, Patuxent, and Patapsco ($r^2=0.16\text{-}0.35$), with the strongest relationship in the Potomac Washington Metro basin (Figure 9-19). These three basins are the ones with the highest percentages of urban land.

As with urban land use in general, Hilsenhoff Biotic Index scores showed a significant increase with low-intensity developed areas, both over all basins (Figure 9-20; $p < 0.001$, $r^2=0.11$) and within the three basins mentioned above. This result again suggests that even a small amount of urbanization may have a significant effect on streams. Hilsenhoff Biotic Index scores were also significantly correlated with high-intensity development, increasing as development increased ($p < 0.001$, $r^2=0.11$).

Statewide, Hilsenhoff Biotic Index scores increased with increasing agricultural land use (Figure 9-21; $p < 0.001$, $r^2=0.02$). This result indicates an increase in degradation with an increased percentage of land in agricultural land use, unlike the results seen with the fish and benthic IBIs. It is likely that the Hilsenhoff Biotic Index is better able to detect organic pollution, a compelling reason for using it as an ancillary indicator to the IBIs. The positive relationship is also seen in six of the individual basins: the Youghiogheny, North Branch Potomac, Upper Potomac, Middle Potomac, West Chesapeake, and Gunpowder ($r^2=0.04\text{-}0.24$), with the North Branch Potomac having the strongest relationship.

Hilsenhoff Biotic Index scores were significantly correlated to the percentage of land in forest cover for all basins, decreasing with increasing forest cover (Figure 9-22; $p < 0.001$, $r^2=0.11$). This significant negative relationship was also noted in eight of the individual basins: the Youghiogheny, North Branch Potomac, Upper Potomac, Middle Potomac, West Chesapeake, Patapsco, Gunpowder, and Chester basins ($r^2=0.04\text{-}0.22$), with the strongest relationship in the Upper Potomac basin.

9.4.4 Associations Between Land Use and Aquatic Salamanders

In addition to the biological indices discussed above, other components of stream communities are significantly affected by land use. Some of these components may prove to be effective new indicators of land use effects; most often the utility of each indicator is dependent on the number and range of values for that indicator. In any case, considering a broader range of biological components can better address impacts on biodiversity.

One promising biological indicator is the number of aquatic salamanders found at each stream site. Although salamander abundance was not included in the results of the 1995-1997 MBSS, fairly reliable counts of aquatic salamander species were obtained. For sites sampled in the 1995-1997 MBSS, the number of aquatic salamander species were plotted against the percentage of catchment area in each land use. Linear regression analyses were conducted to evaluate the strength of associations between land use and biological condition. Although the number of aquatic salamanders per stream site never exceeded five, aquatic salamander richness was significantly correlated with the percentage of urban, agricultural, and forest land uses.

Statewide, the number of aquatic salamander species decreased with increasing urban land use, indicating a loss of biodiversity with more urban land (Figure 9-23; $p < 0.0001$, $r^2=0.03$). This relationship was also significant for aquatic salamander species richness in the Highlands ($p < 0.017$, $r^2=0.02$) and Piedmont ($p < 0.0002$, $r^2=0.04$) regions of Maryland. A similar negative relationship was observed between aquatic salamander species richness and increasing agricultural land use statewide ($p < 0.0038$, $r^2=0.01$) and in the Highlands ($p < 0.0001$, $r^2=0.05$). A significant positive relationship was evident in the Piedmont, likely reflecting the reciprocal relationship between agriculture and urban uses in that region. As expected, aquatic salamander species richness increased with increasing forested land use statewide ($p < 0.0001$, $r^2=0.05$) and in the Highlands ($p < 0.0001$, $r^2=0.07$). The relationship in the Piedmont was not significant.

Especially in small streams that often contain few or no fish species, aquatic salamanders appear to be an effective indicator of land use influences. Unlike fish, aquatic salamanders showed a negative association with agricultural land use statewide. Future monitoring efforts may improve this indicator by adding abundance measures and more thoroughly sampling for adult and larval salamanders.

9.4.5 Associations Between Land Use and the Physical Habitat Index

Although linkages between watershed land use and physical habitat conditions have been demonstrated in a number of studies, MBSS statewide results did not indicate declines in PHI scores with increased urban or agricultural land use. It is likely that the parameters included in the PHI do not represent all the aspects of habitat quality that can be affected by human alterations to watershed lands. Further examination of individual habitat factors might reveal stronger associations with catchment land use.

Within several individual basins, some associations between land use and the PHI were detected. In the Potomac Washington Metro basin, agricultural land use had a significant negative relationship with PHI ($p=0.002$, $r^2=0.14$). Forest land cover had a significant positive relationship with PHI in the Potomac Washington Metro ($p=0.01$, $r^2=0.09$) and Bush ($p=0.03$, $r^2=0.26$) basins.

The lack of correspondence between land use and PHI is not unexpected, given the scale of analysis. Certainly, some processes that affect physical habitat do operate on a watershed level: for example, sediment transport may increase embeddedness and flow variability leads to channel instability and degradation of naturally-occurring riffles and pools. However, other components of physical habitat are affected or assessed at a more local scale. The amount of instream woody debris at a particular site depends on the availability of nearby tree cover. Maximum depth depends on watershed size and local variation in geography, although in some cases major flow fluctuations can result in development of shallow, overwidened channels. Aesthetic quality is assessed at a local level, based on streamside field observations. Thus a stream in a forested park, within an otherwise developed watershed, may still rate high in aesthetic quality. Clearly, numerous aspects of physical habitat quality are affected by land use, although not always in ways detected by our GIS-based estimates.

9.5 IMPLICATIONS

In general, biological indicators did show a number of significant relationships to catchment land uses. Fish and benthic IBI scores were particularly sensitive to the degree of watershed urbanization, but were less able to detect effects of agriculture at the watershed scale. Benthic IBI scores increased with the amount of forest cover. The Hilsenhoff Biotic Index was able to detect degradation associated with both urban and agricultural lands, and was also related to forest cover. In many cases, examining relationships within individual basins provided a clearer picture of land use relationships than did statewide results.

Urbanization and agriculture have historically exerted and will continue to exert significant pressure on stream ecosystems in Maryland. Currently, three basins (Patapsco, Potomac Washington Metro, and West Chesapeake) contain the majority of sampled sites with greater than 25% urban land in the upstream catchment. However, as human population continues to grow, development pressure (and with it, the percentage of urban land) will likely extend to other parts of the state. Recent statewide efforts to improve land use planning and requirements for stormwater management may lessen the negative impacts of urban and suburban development. Programs aimed at reducing point and nonpoint nutrient loadings to the Chesapeake Bay (such as Maryland's Tributary Strategies, riparian reforestation, and management of crop nutrients and animal waste) will likely benefit streams as well.

While this analysis represents significant progress in understanding the ecological effects of urbanization and agriculture at the statewide and river basin scales, additional studies will likely provide further insights. The extent of agricultural influence does not take into account variations in land slope, soil erodibility, or implementation of Best Management Practices that may exacerbate or ameliorate adverse effects at individual sites. Similarly, urban impacts may vary, depending on the amount of impervious surface and the nature of point sources discharging to streams. Perhaps most importantly, the composition of riparian (streamside) land cover is critical to understanding the influence of land use and to target conservation measures (such as reforestation) that can improve stream conditions. Related studies are now underway to compare the influences of riparian and catchment conditions, using MBSS data for the Patapsco and other basins. Other efforts are continuing to improve on existing predictive models by incorporating other indicators of landscape condition (e.g., impervious surface), as well as other stressors (see Chapter 11).

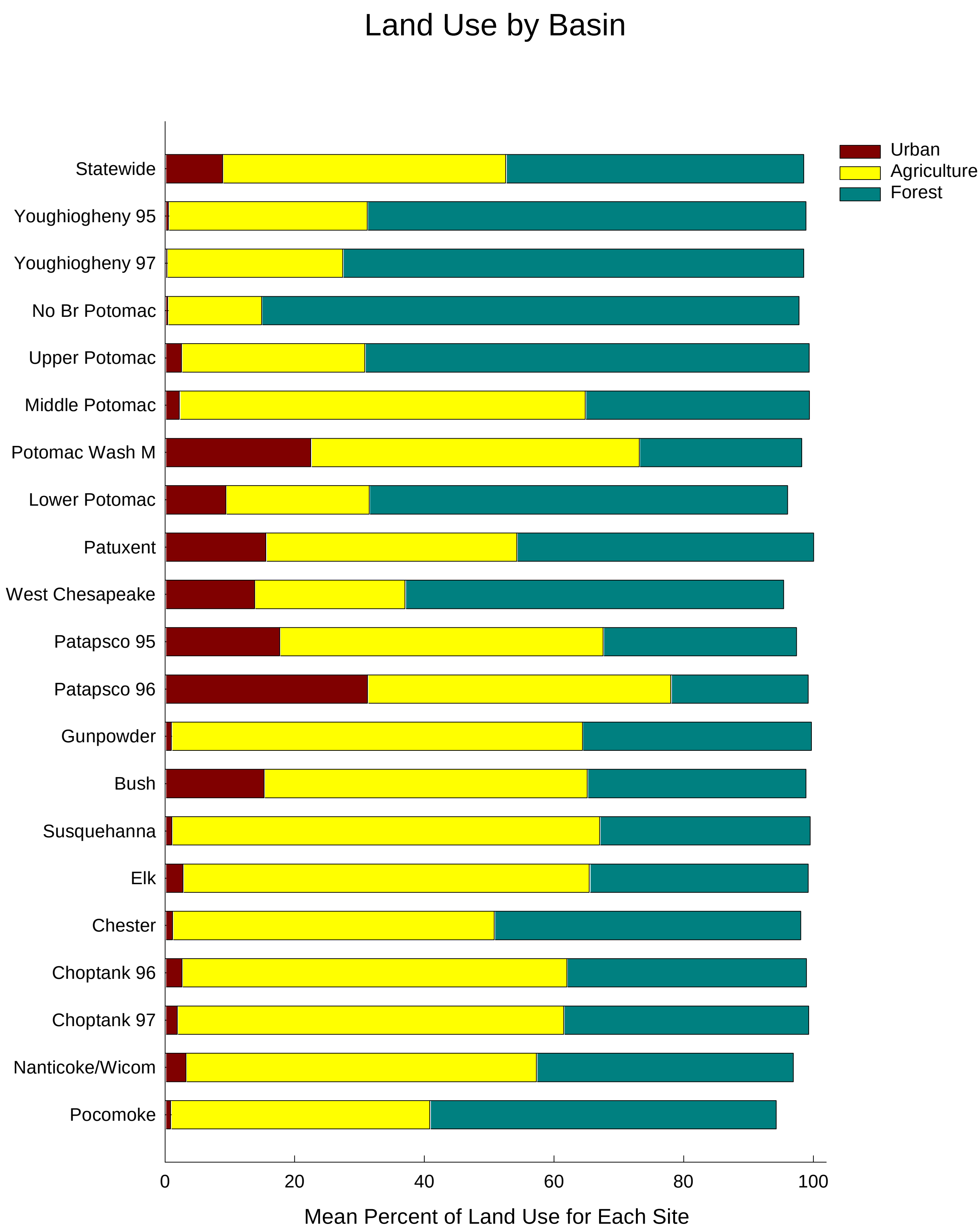
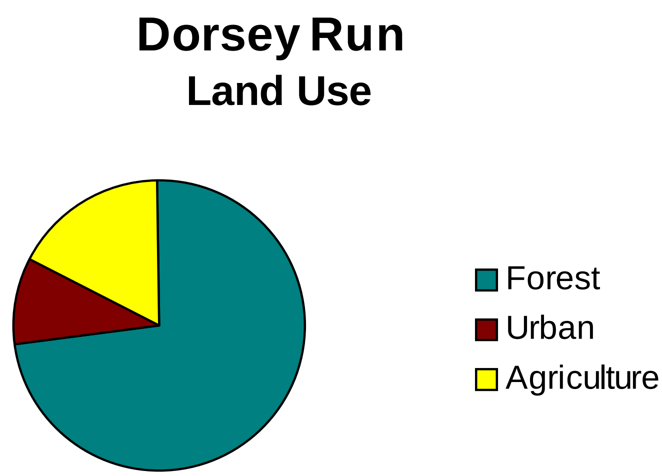


Figure 9-1. Major land use types within individual catchments upstream of the 1995-1997 MBSS sampling sites. Values for each basin are the mean percentage of catchment area in each of the land use categories.



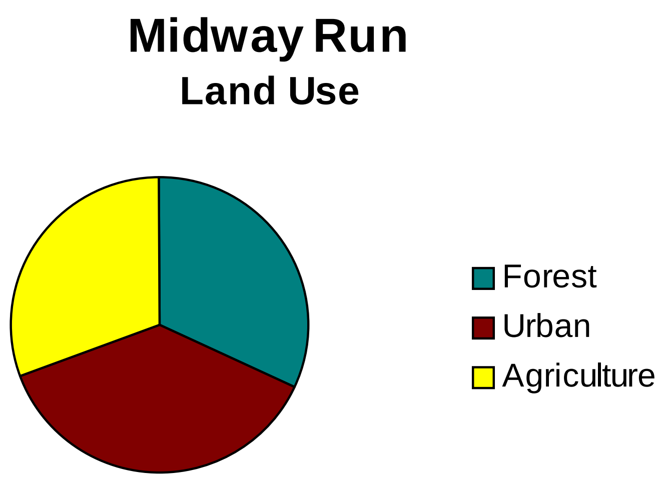


Figure 9-2. Percentage of three land use types (forest, urban, and agriculture) for two streams in the Patuxent basin - Dorsey Run and Midway Run

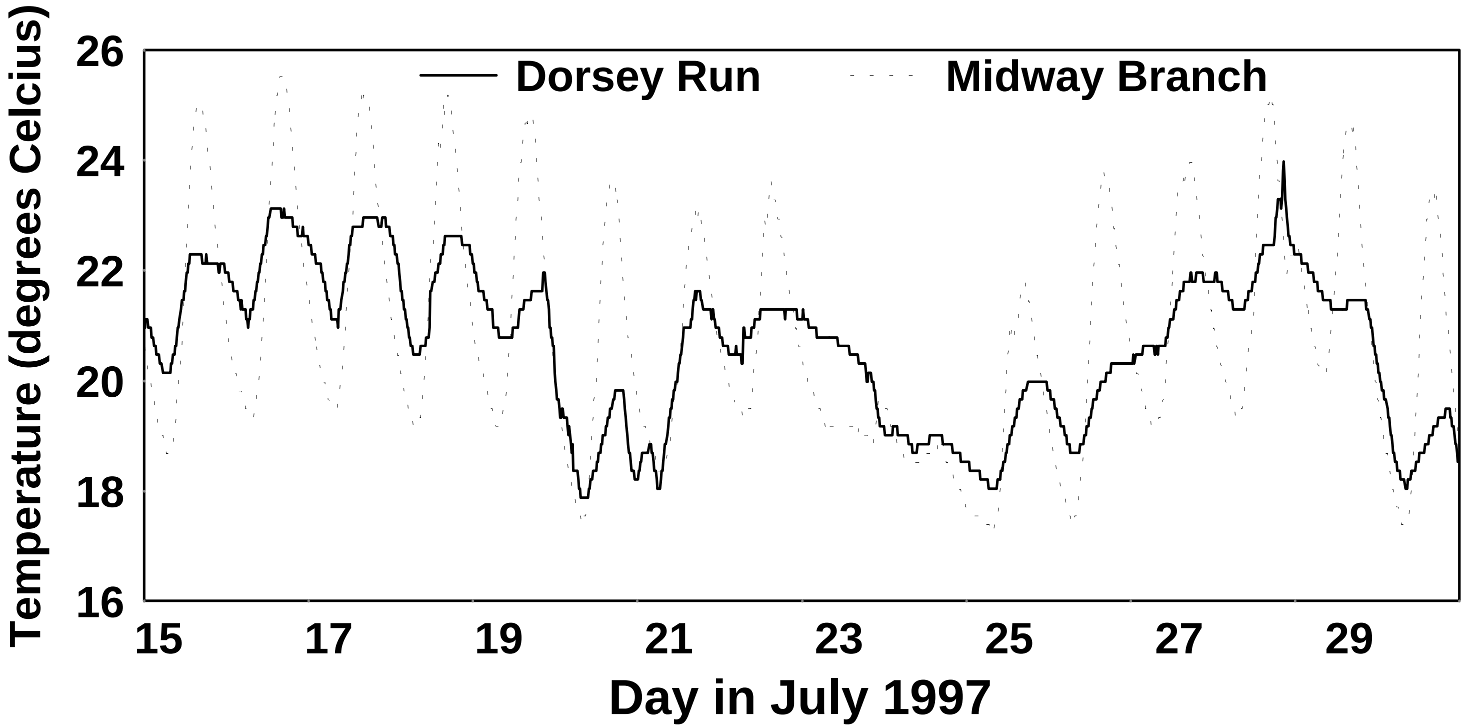


Figure 9-3. Water temperature (◦C) during July 1997 for two streams in the Patuxent basin - Dorsey Run and Midway Run

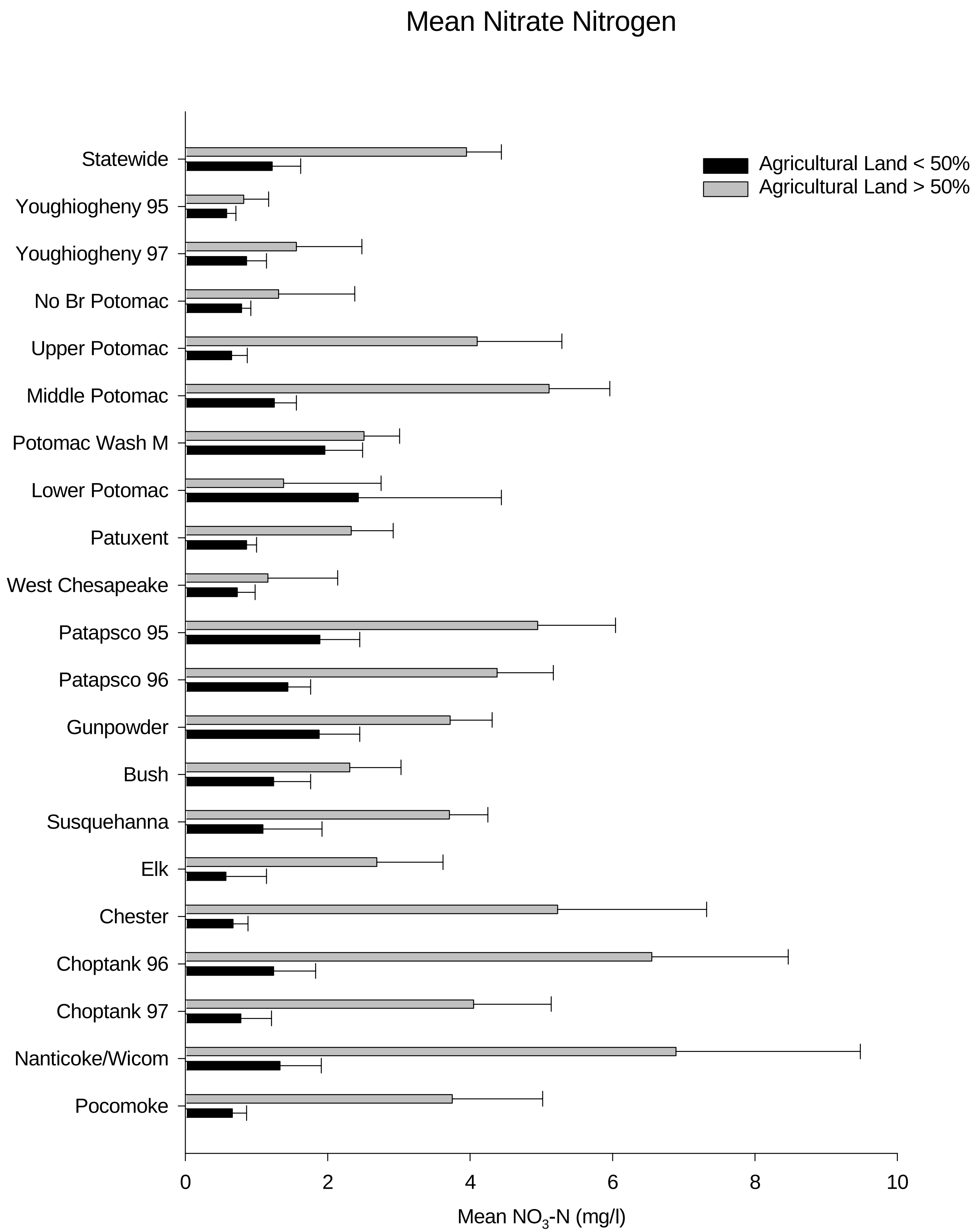


Figure 9-4. Mean nitrate-nitrogen concentration (mg/l), statewide and for basins sampled in the 1995-19979 MBSS, among sites with catchment land use less than and greater than 50% agriculture

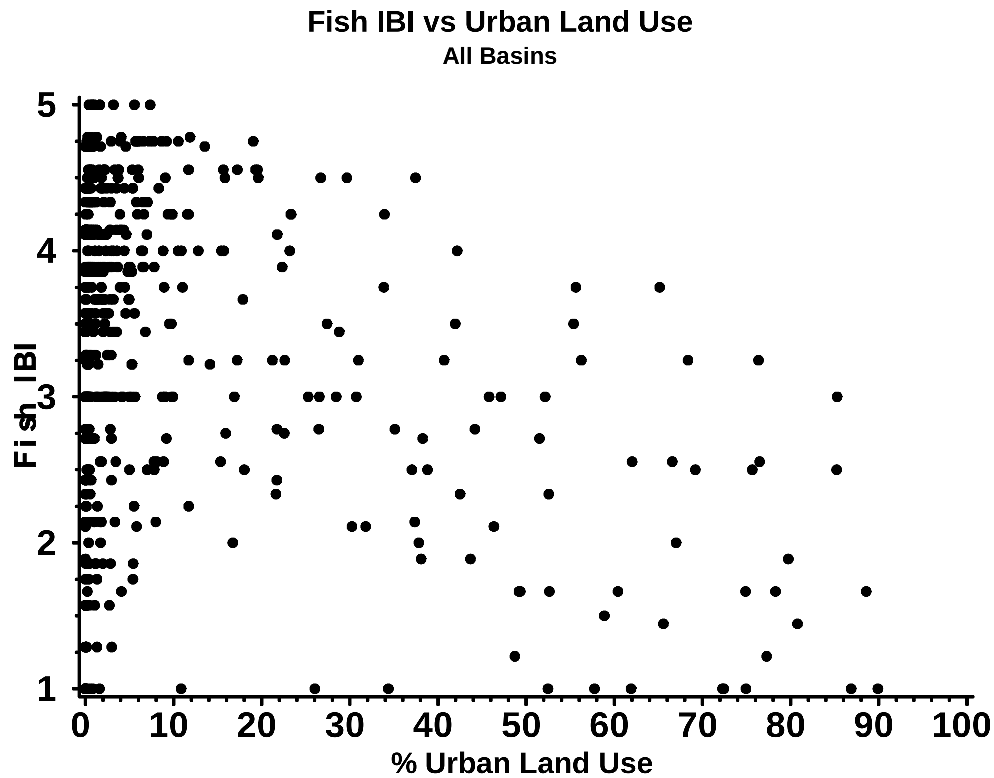


Figure 9-5. Relationship between the fish IBI and urban land use for the basins sampled in the 1995-1997 MBSS

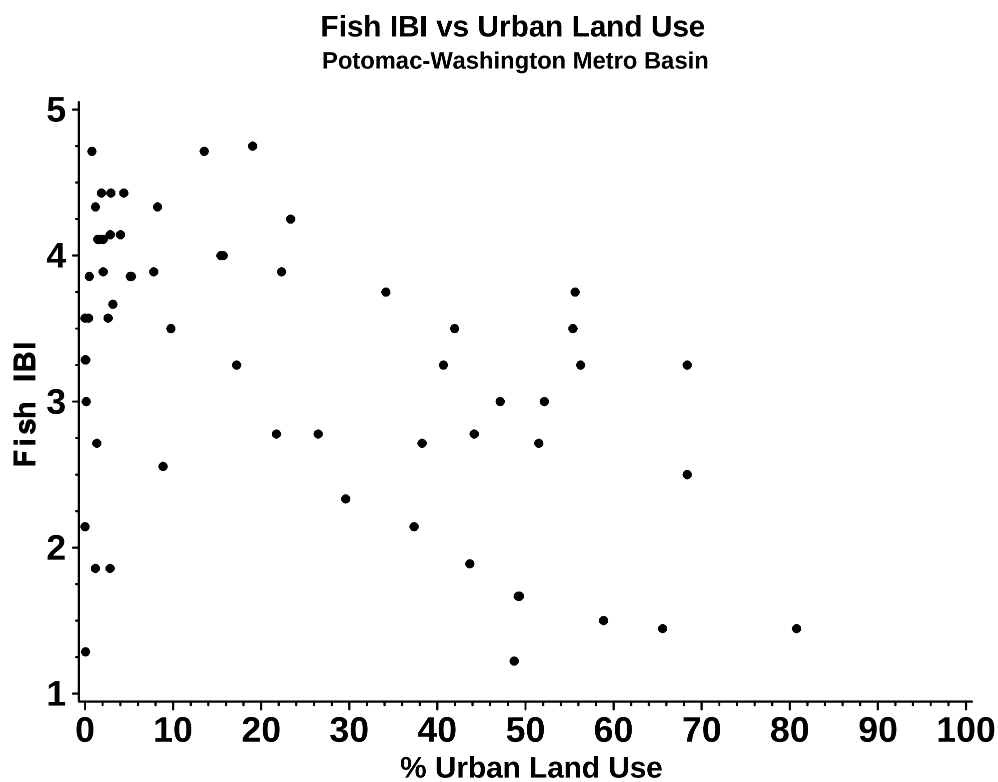


Figure 9-6. Relationship between the fish IBI and urban land use for the Potomac Washington Metro basin

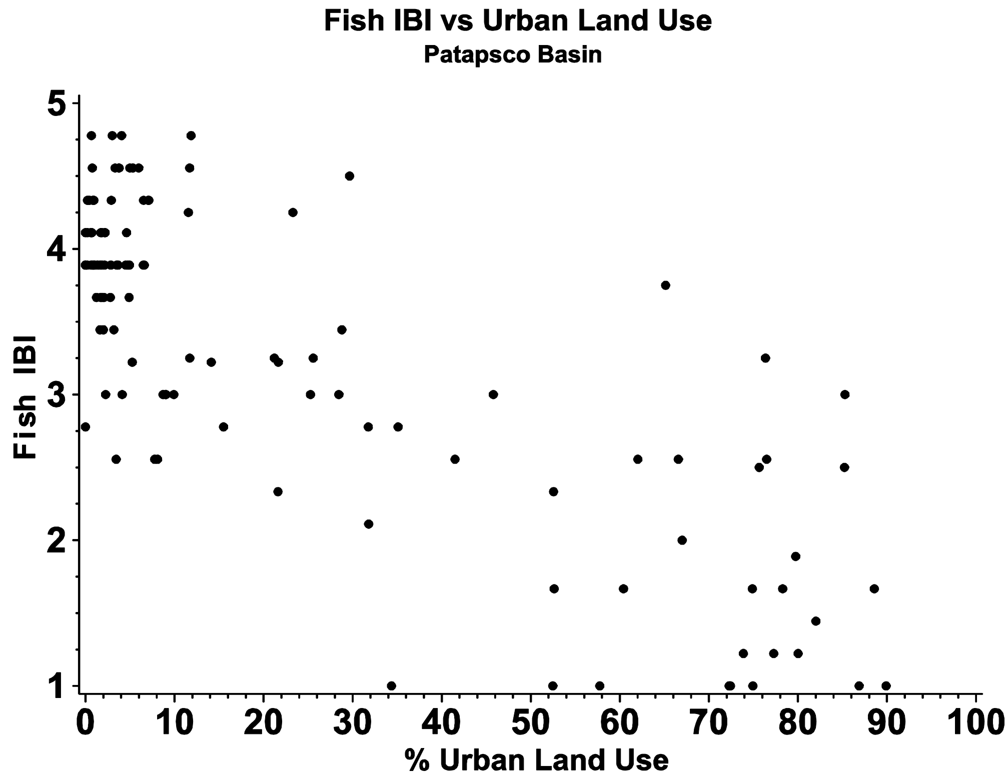


Figure 9-7. Relationship between the fish IBI and urban land use for the Patapsco basin

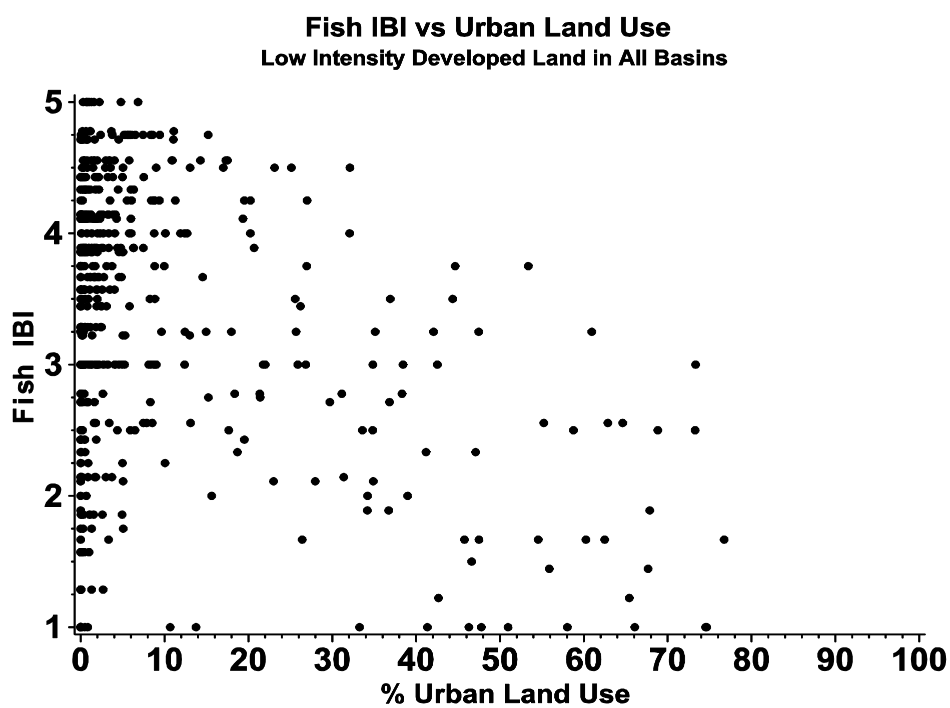


Figure 9-8. Relationship between the fish IBI and low-intensity development for the basins sampled in the 1995-1997 MBSS

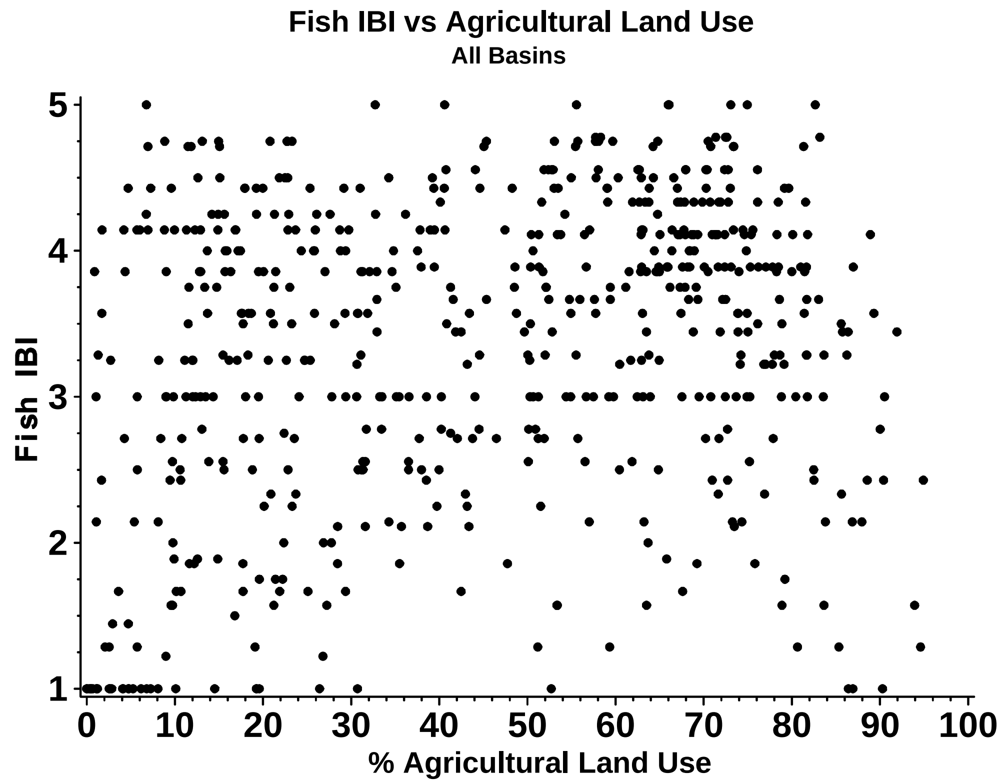


Figure 9-9. Relationship between the fish IBI and agricultural land use for the basins sampled in the 1995-1997 MBSS

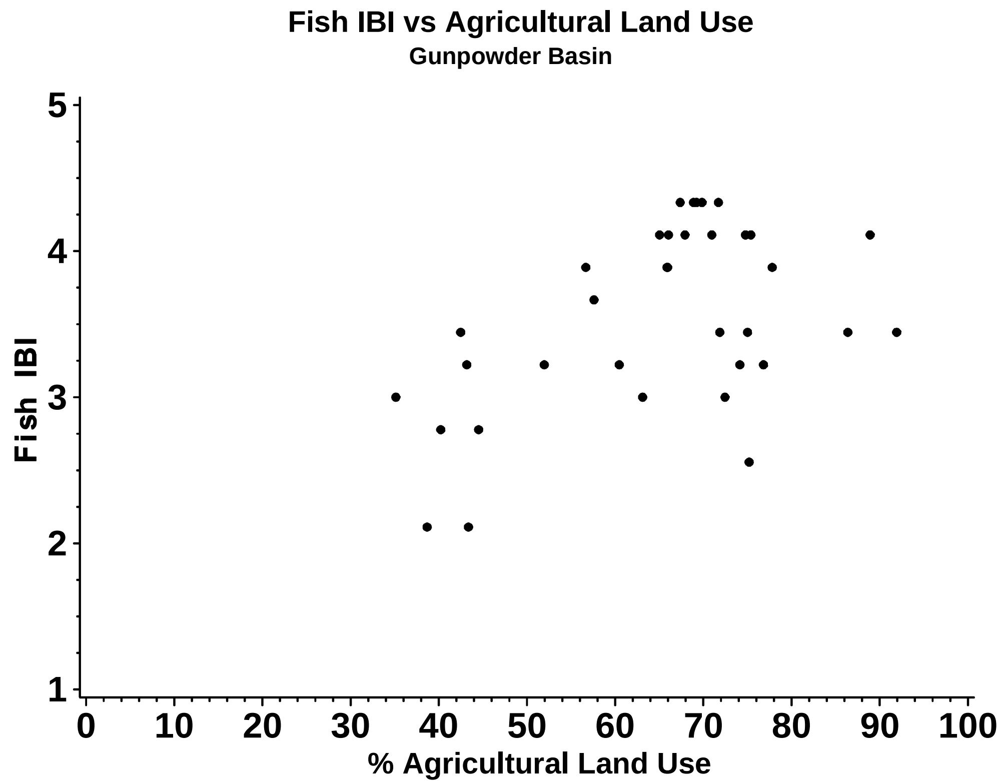


Figure 9-10. Relationship between the fish IBI and agricultural land use for the Gunpowder basin

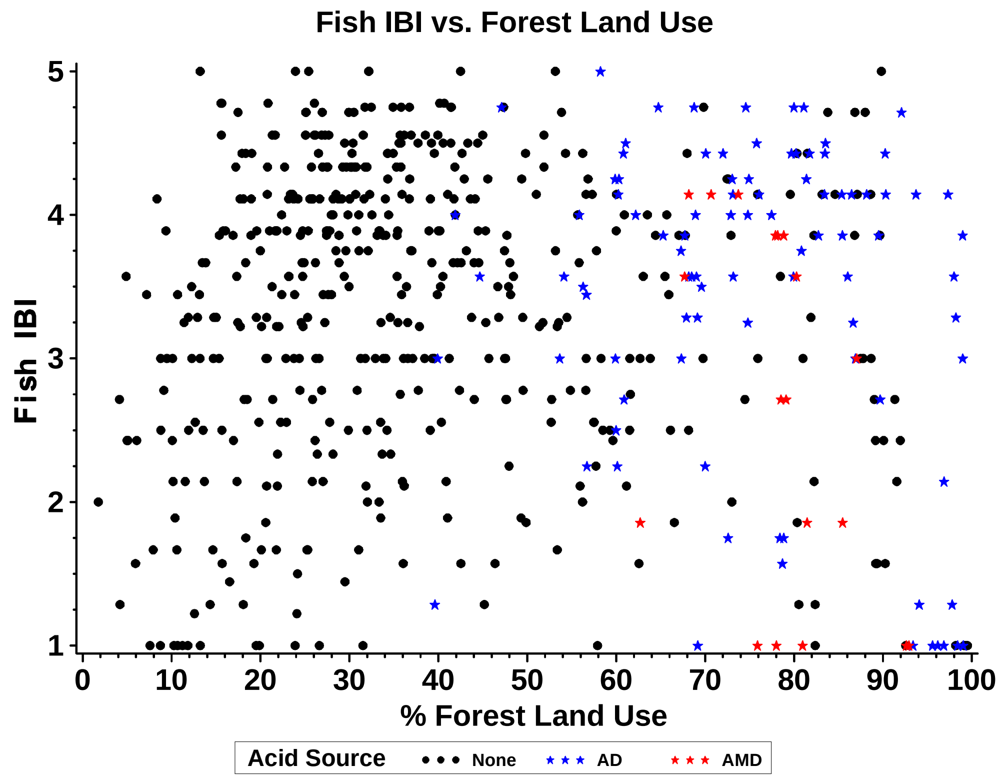


Figure 9-11. Relationship between the fish IBI and forested land cover for the basins sampled in the 1995-1997 MBSS. Blue stars indicate sites affected by acid deposition (AD); red stars indicate acid mine drainage (AMD).

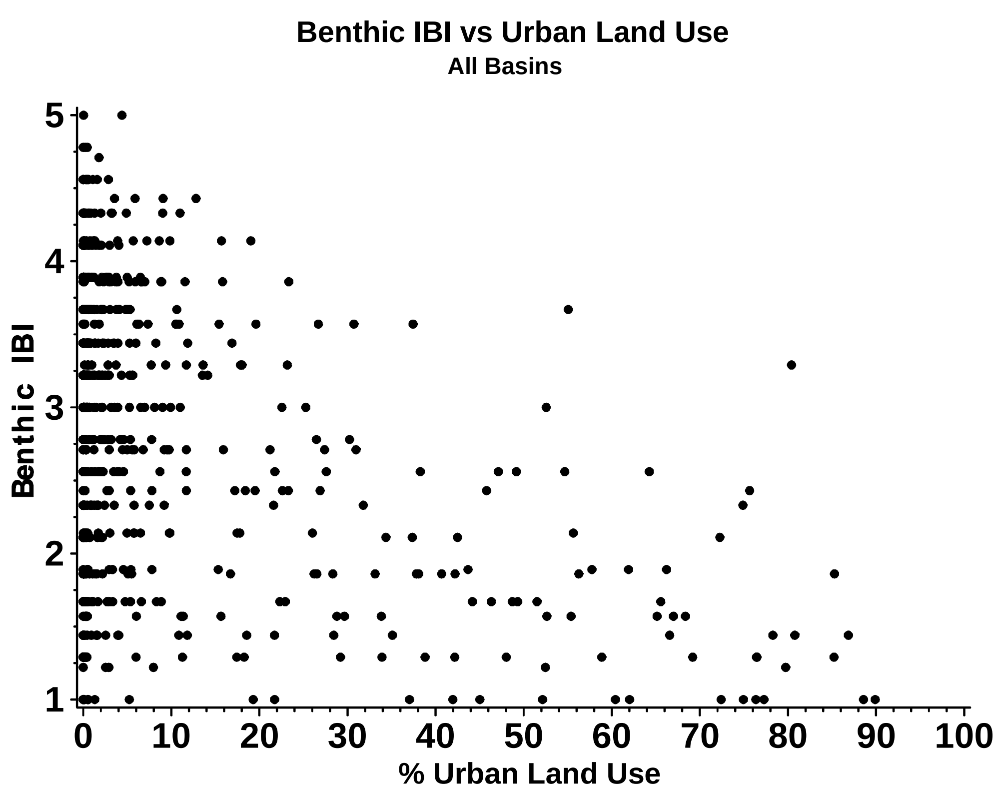


Figure 9-12. Relationship between the benthic IBI and urban land use for the basins sampled in the 1995-1997 MBSS

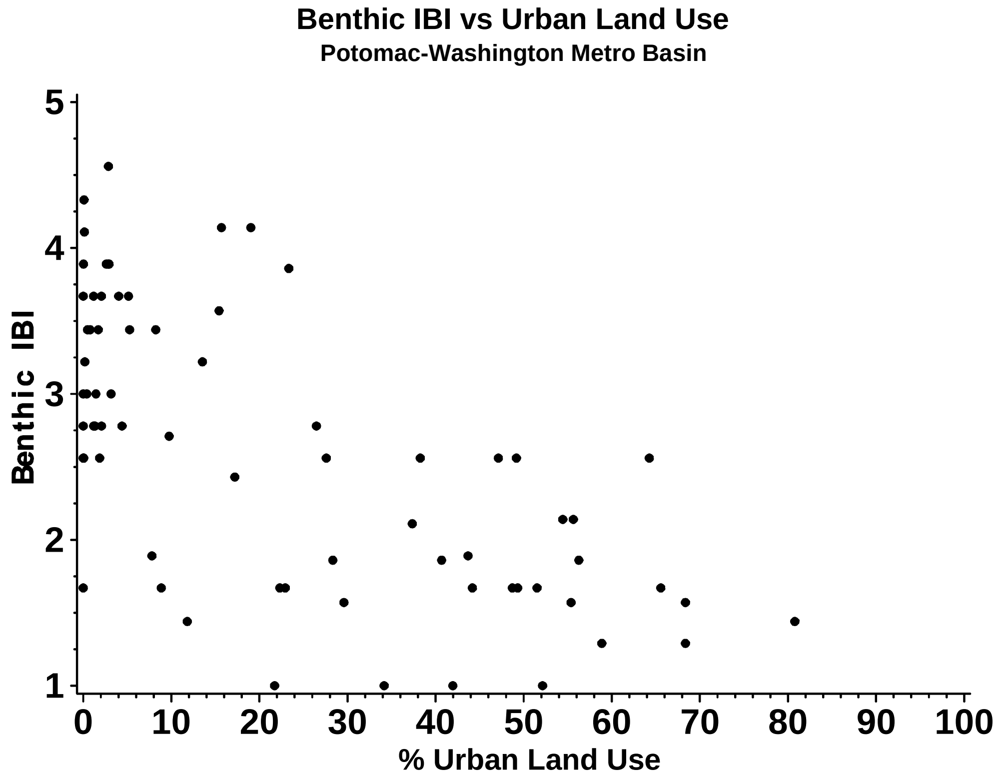


Figure 9-13. Relationship between the benthic IBI and urban land use for the Potomac Washington Metro basin

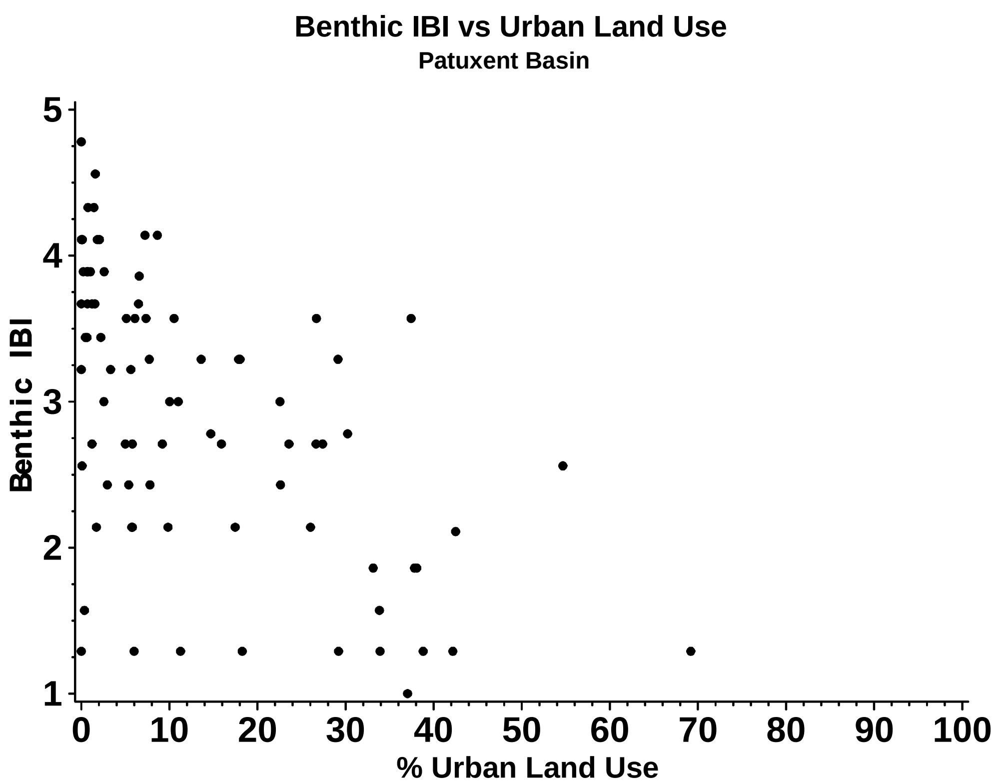


Figure 9-14. Relationship between the benthic IBI and urban land use for the Patuxent basin

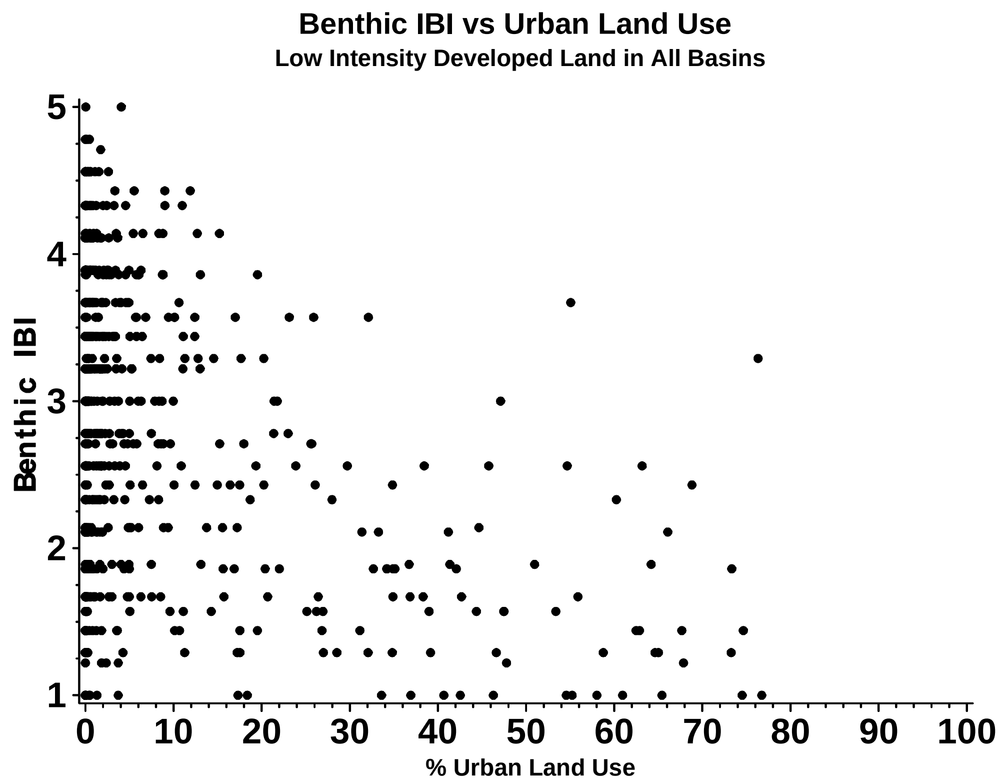


Figure 9-15. Relationship between the benthic IBI and low-intensity development for the basins sampled in the 1995-1997 MBSS

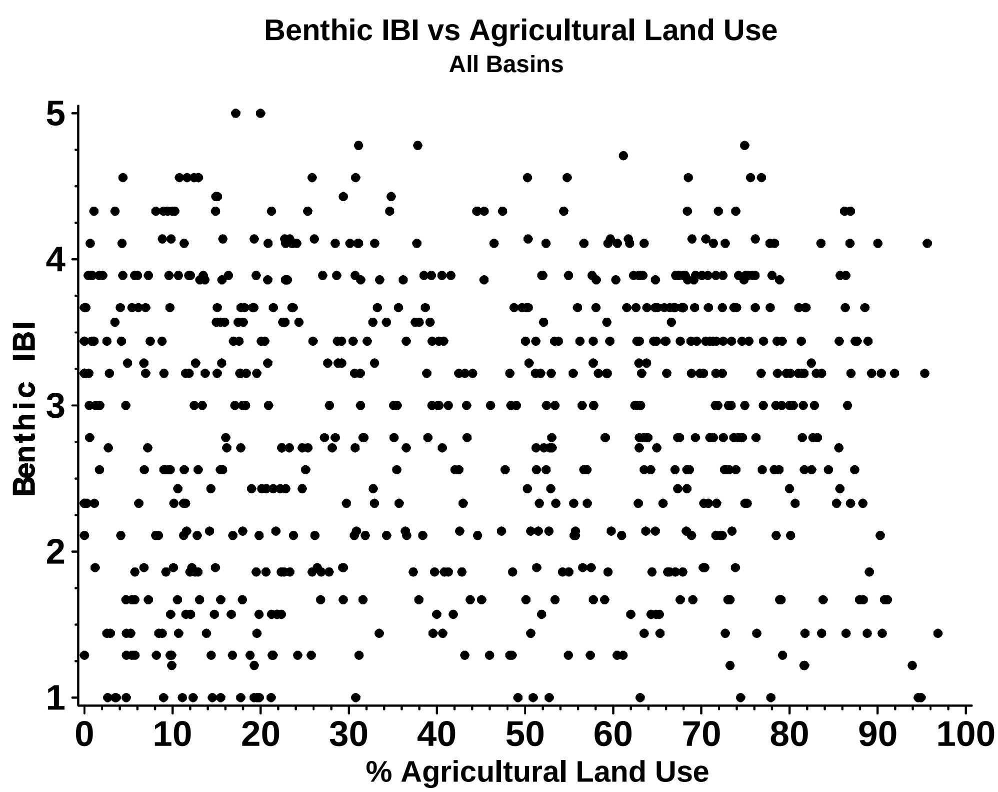


Figure 9-16. Relationship between the benthic IBI and agricultural land use for the basins sampled in the 1995-1997 MBSS

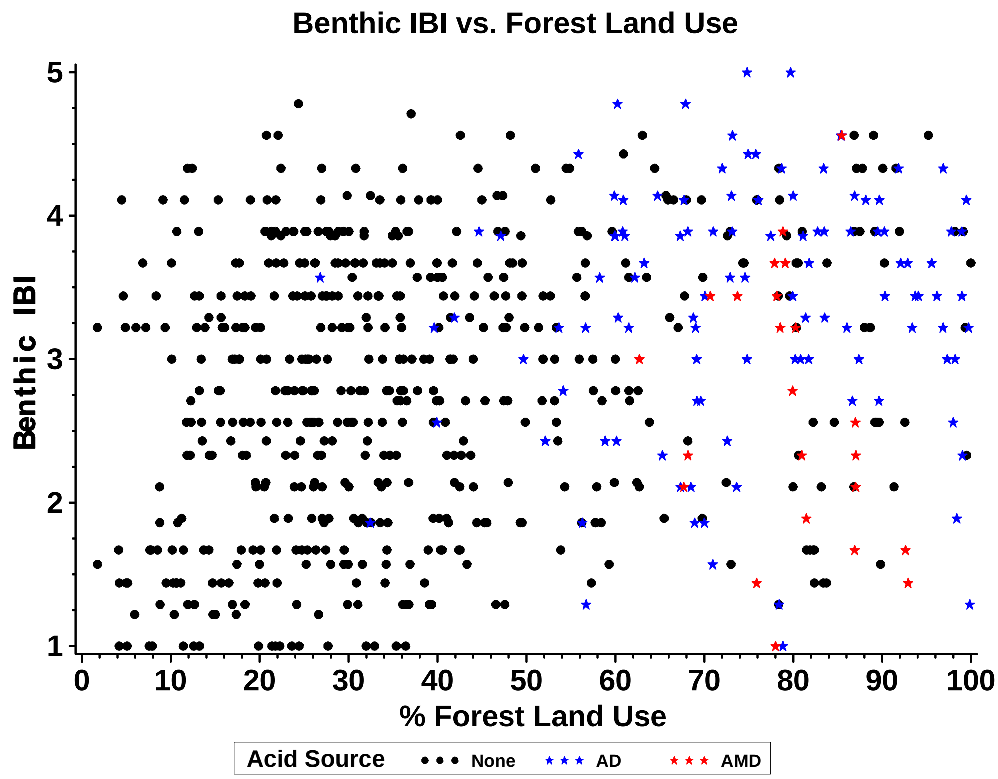


Figure 9-17. Relationship between the benthic IBI and forested land cover for the basins sampled in the 1995-1997 MBSS. Blue stars indicate sites affected by acid deposition (AD); red stars indicate acid mine drainage (AMD).

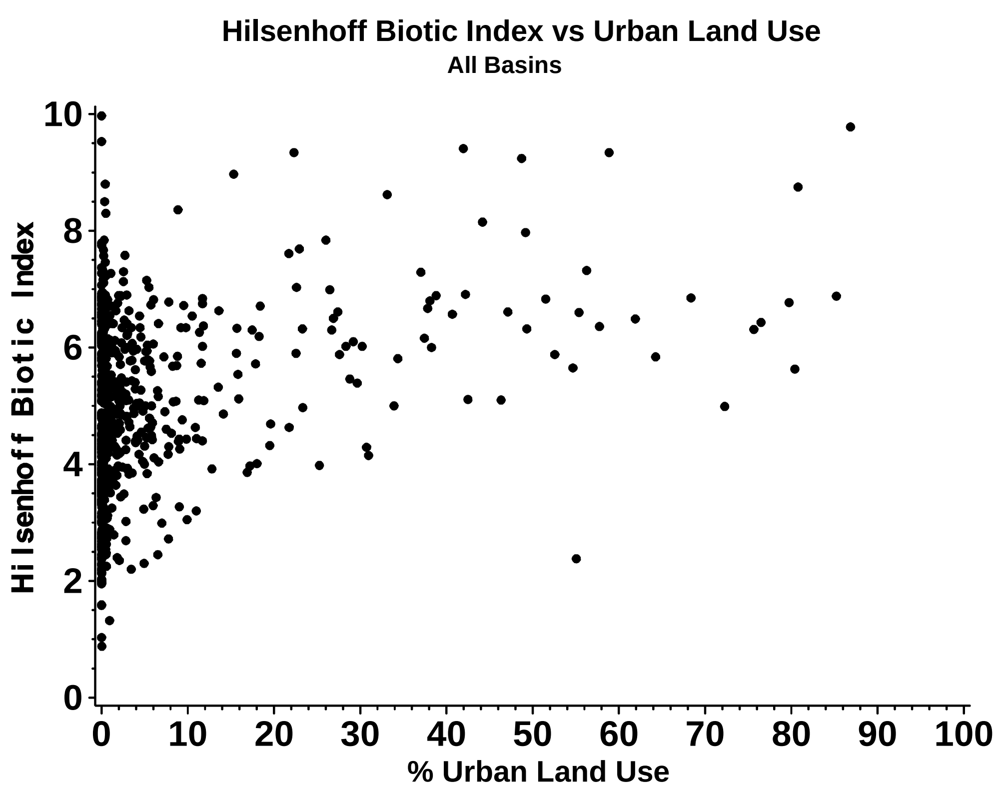


Figure 9-18. Relationship between the Hilsenhoff Biotic Index and urban land use for the basins sampled in the 1995-1997 MBSS

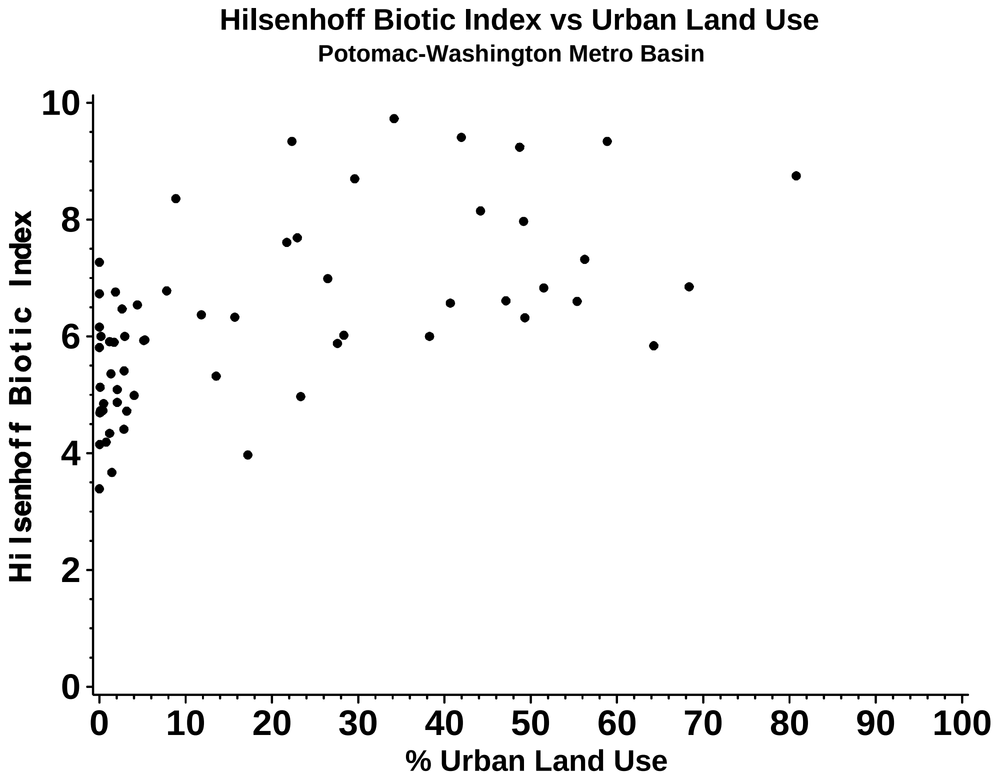


Figure 9-19. Relationship between the Hilsenhoff Biotic Index and urban land use for the Potomac Washington Metro basin

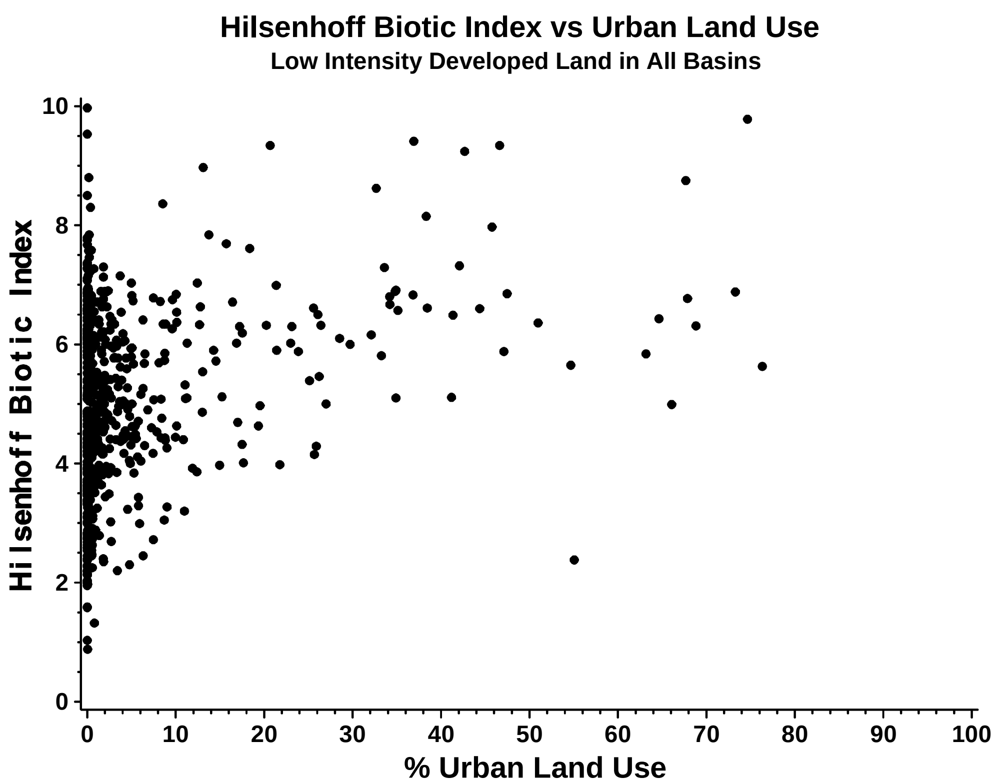


Figure 9-20. Relationship between the Hilsenhoff Biotic Index and low-intensity development for the basins sampled in the 1995-1997 MBSS

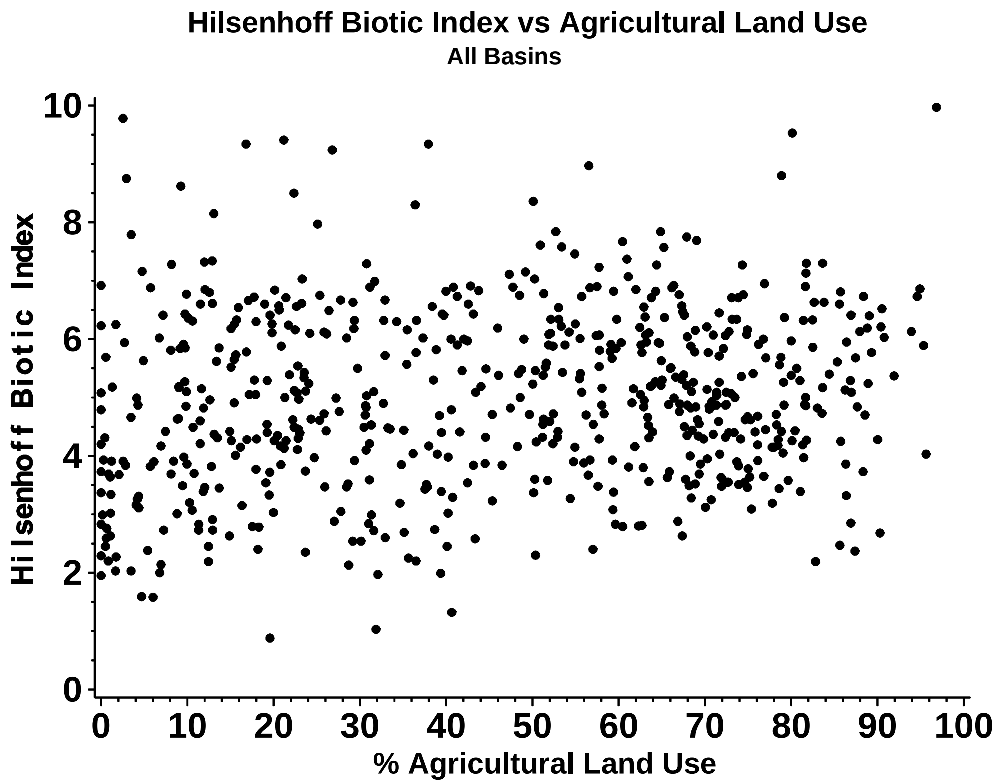


Figure 9-21. Relationship between the Hilsenhoff Biotic Index and agricultural land use for the basins sampled in the 1995-1997 MBSS

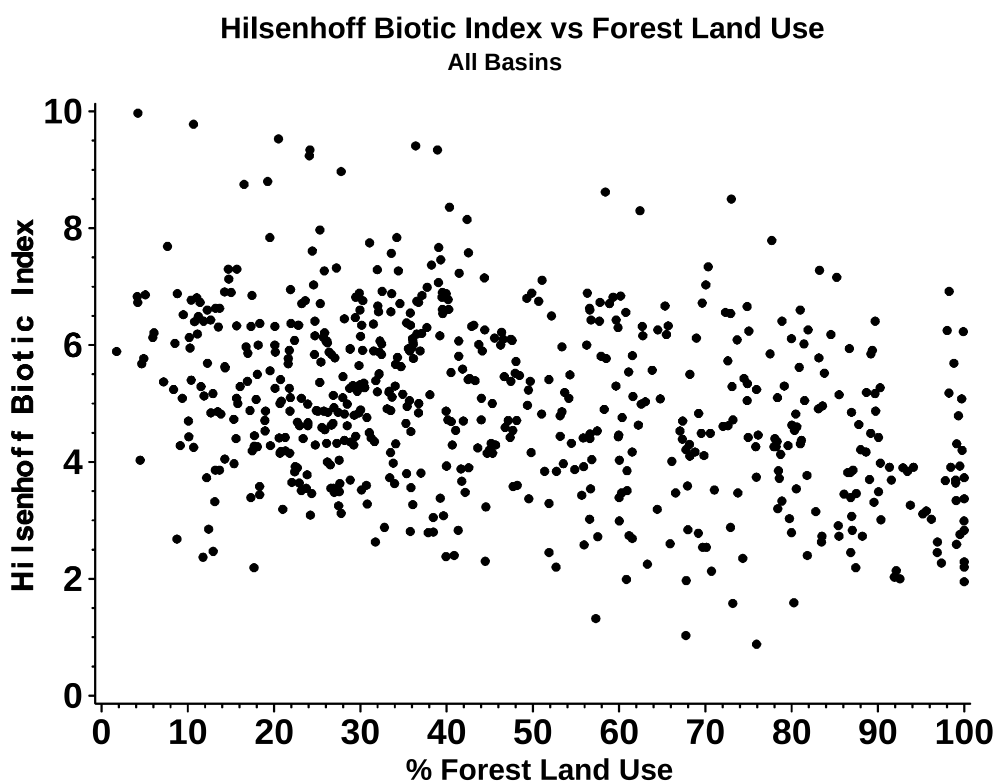


Figure 9-22. Relationship between the Hilsenhoff Biotic Index and forested land cover for the basins sampled in the 1995-1997 MBSS

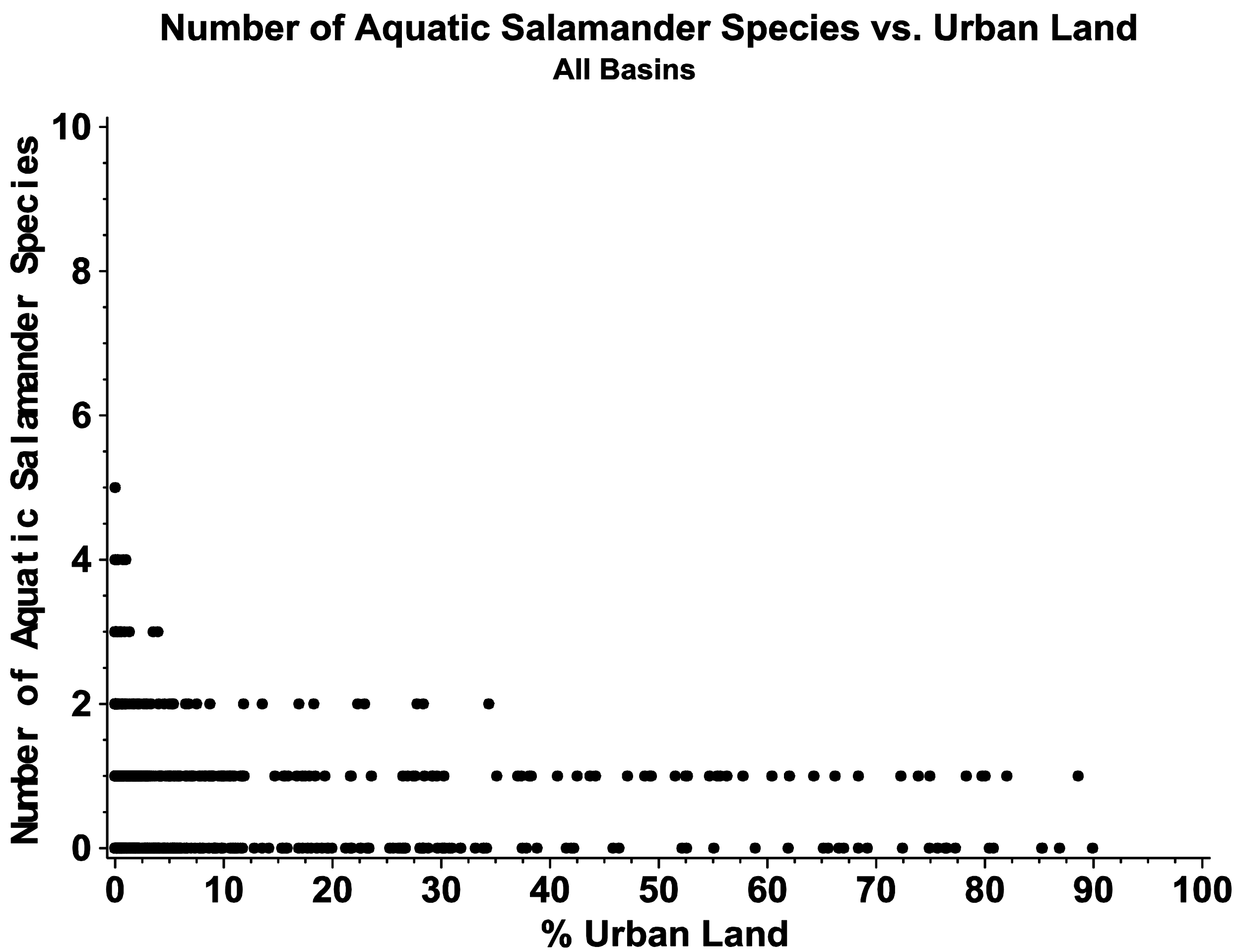
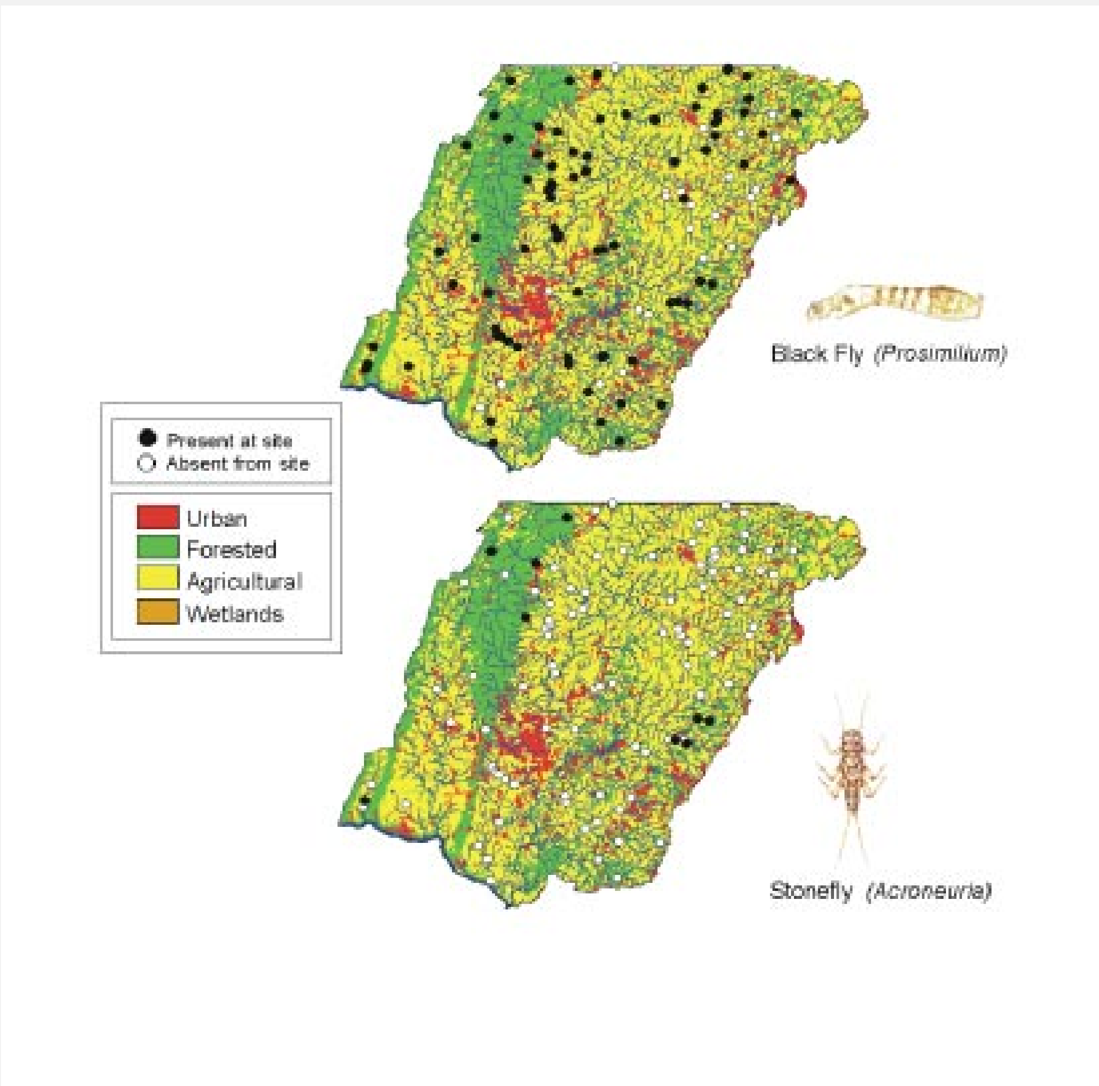


Figure 9-23. Relationship between the number of aquatic salamanders per site and urban land use for the basins sampled in the 1995-1997 MBSS

Benthic Taxa as Indicators of Stream Degradation

The presence or absence of certain benthic macroinvertebrate taxa can indicate the effects of watershed land uses. For example, the stonefly *Acroneuria* is pollution-sensitive and survives only among clean rocks in streams with cool, swiftly-moving water and a good amount of dissolved oxygen. In the Middle Potomac River basin, which is mostly agricultural land, these insects were found at only 9 of the 109 sites sampled and primarily in the heavily-forested mountains in the western part of the basin. Streams here are likely to be less polluted by sediment, nutrients, pesticides, and herbicides that often enter streams in runoff from agricultural areas. However, the more pollution-tolerant black fly , *Prosimulium,* was found throughout the basin - in forested, agricultural, and urban watersheds. These insects can live in degraded streams in the more developed areas of the basin. Combined influences of land uses on the entire benthic community – changing the relative abundance of tolerant and sensitive species – are reflected in community-based indicators such as the benthic IBI.



In the Middle Potomac basin, sensitive *Acroneuria* stoneflies were found in less-disturbed streams, while tolerant *Prosimilium* tolerated a wide range of land use conditions.

Amphibians and Reptiles Sensitive to Urbanization

A number of amphibians and reptile species appear to be particularly sensitive to the effects of urban development. Of the 29 aquatic or ripa*rian speci*es of amphibians and reptiles found during the survey, only seven occurred in heavily-urbanized areas (>25% imperfious land cover in the upstream watershed). At the opposite end of the scale, four species of salamanders (in blue) never occurred in urbanized areas (>3%) impervious land cover).